LCA Methodology

Country-specific Damage Factors for Air Pollutants
A Step Towards Site Dependent Life Cycle Impact Assessment

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Abstract. An integrated impact assessment model is used to calculate the impact per tonne of SO_2, NO_x, fine particles, and NMVOC emitted from different source countries on human health, acidification, eutrophication, and the man-made environment (crop yield and building materials). Indicators on the endpoint level are used to measure the effects resulting from a marginal change in emission levels. While the assessment of impacts on ecosystems and the man-made environment is limited to Europe, damage factors for health effects are also derived for Asia and South America. For Europe, emission scenarios for the years 1990 and 2010 are considered to analyse the influence of changing background conditions on the resulting impacts. Results show that there is a significant variation in the damage resulting from a unit emission for some of the impact categories, both between countries and between base years. Depending on the scope of the study and the information available from the life cycle inventory, results from the paper can be used to consider site dependent conditions in life cycle impact assessment as a complement to the current site-independent (or global) approach.

Keywords: Acidification; air pollution; eutrophication; integrated impact assessment modelling; Life Cycle Assessment; Life Cycle Impact Assessment; site dependent impact assessment; Years of Life Lost

Introduction

There is an emerging discussion on if at all, how, and to which extent site dependent conditions should be accounted for in life cycle impact assessment. While for both methodological and practical reasons the release of a substance, emitted either from a ship in the middle of the Atlantic or in the centre of a highly populated urban area, is treated equally in 'traditional' LCIA practice, there is a growing demand, partly supported e.g. by the SETAC Working Group on Life Cycle Impact Assessment (Udo de Haes 1999) for addressing potential site dependent parameters in a more appropriate way.

According to Guinèe et al. (1998), there are currently three main approaches to regionalisation in LCIA:

1. Distinction between sensitive and non-sensitive areas. Pollutants released in a clearly non-sensitive area are not considered.

2. Introduction of an effect-oriented site factor, which varies between 0 and 1 according to the expected sensitivity of the area in which the substance is emitted.

3. Introduction of site dependent characterisation factors based on fate and effect modelling.

As site dependent conditions in general influence both fate and exposure processes as well as the effect mechanism, only approach (3) seems suitable to capture site dependent effects in an appropriate way, but at the same time it is obviously the most resource intensive. Approaches (1) and (2) take into account site dependent parameters for the effect assessment, but do not consider fate and exposure as site dependent, which might be appropriate in specific cases, but is not generally valid.

Potting et al. (1998) and Huijbregts (1999) have used an integrated impact assessment model (RAINS) to derive site dependent acidification factors. In the present paper we use a similar type of model (EcoSense) to calculate site dependent damage factors for a selected set of air pollutants and related impact categories (human health, acidification, terrestrial eutrophication, man-made environment). Bearing in mind the limited number of substances that we are currently able to cover with the models described below, apart from providing a set of site dependent damage factors a perhaps evenly important objective of this work is to contribute to the current discussion on the usefulness of spatial differentiation in LCIA. Which are the impact categories that are particularly sensitive to site specific conditions? Do the differences between site dependent damage factors matter at all? What are the driving parameters for these differences? We think that the results presented below give some new insights into these issues.

1 Methodology

1.1 General framework

The objective of the analysis is to quantify some of the effects resulting from a unit release of a substance from specific regions. For the quantification of effects we use as far as possible indicators on the endpoint level, i.e. at the level which directly matters to society. Impact indicators on the midpoint level (i.e. indicators closer to the environmental intervention) in general are less appropriate to analyse the influence of site dependent conditions.

To analyse the relation between a unit emission of a given substance at a specific site and the resulting effects, we use the modelling framework described in section 2. The analysis is primarily focused on emissions in Europe (EU-15 countries), but we also use a first version of the EcoSense model for Latin America and Asia to compare health related damage factors between continents. To calculate site dependent damage factors, in a number of consecutive model runs we respectively increased the total national emissions of the respective pollutant for individual countries by 10%, an approach that was applied also by Potting et al. (1998) to calculate country-specific acidification factors. The reasoning for this procedure is as follows:

- *Change of emissions on the country level:* The modelling of fate, exposure and effect is based on a regular grid that provides a higher spatial resolution (50 x 50 km) and thus a higher level of accuracy than the country level. However, the allocation of an emission source to a specific grid cell or to longitude and latitude coordinates is – at best – not very practical for the LCA practitioner. The allocation of an emission source to a country is not necessarily very precise, as in particular for large countries the environmental conditions might differ significantly between different sites in the same country, but the use of national average values at least in Europe seems to be a reasonable pragmatic approach, as it keeps the allocation to a site operational. It might be mentioned here also that the model takes into account the actual spatial distribution of emission sources within the country, i.e. it does not assume the emission load to be evenly distributed across the country.
- *Increase of national emissions by 10%:* Although the change of emissions by 10% is somehow arbitrary, we consider this approach as a pragmatic compromise between a marginal and average analysis. For some of the impact mechanisms we look at, the working point of the fate-exposure-effect mechanism is determined by site dependent background conditions, so that marginal changes are required to estimate site dependent impact factors. A 10% change in emissions is considered to not deviate too far from the working point, and at the same time allows a certain generalisation of results.

To analyse the influence of background conditions in more detail, the 10% change of emissions from the individual EU countries is applied to two different base years, namely 1990 and 2010, which are characterised by different emission levels, and thus different background concentrations in the atmosphere. The European emission inventory for 1990 is taken from the CORINAIR database (McInnes 1996). For the year 2010 we use an emission inventory defined in a IIASA study (Amann et al. 1999) by taking into account current national and international legislation and emission control strategies that will affect emission levels in 2010. On the EU-15 average, in the year 2010 the SO_2 emissions are reduced by 71%, NO_x emissions by 48%, VOC emissions by 49%, and NH_3 emissions by 12% compared to 1990 emission levels. For simplicity reasons we assumed population and land use patterns to remain constant over time.

In addition to the analysis of country-specific damage factors we have used the same methodology to explore whether there

are also systematic differences between the effects per unit emission from different industry sectors, which might result from a specific spatial pattern of industrial activities in Europe. As the CORINAIR emission inventory, which is used here as a basis for the definition of European emission scenarios, takes into account the actual spatial distribution of emission sources in Europe, we can use our model to calculate damage factors for specific source sectors (e.g. energy sector), which take into account site dependent conditions at the respective source sites in Europe. CORINAIR differentiates between eleven main source sectors, which are disaggregated to more than 200 different emitting activities. Similar to the approach followed for the calculation of country-specific damage factors, in a series of model runs we increased emissions from the CORINAIR main source categories in the EU-15 countries by 10% respectively. To account for the difference in the stack height, the model distinguishes between sources with a stack height below and above 100 m.

1.2 Impact categories and pollutants considered

The effects from the emission of SO_2, NO_x, fine particles and volatile organic compounds (VOC) on human health, acidification, terrestrial eutrophication, crop yield, and building materials are assessed. The following sections provide a brief description of the impact indicators and the basic assumptions underlying the impact assessment.

1.2.1 Human health

The indicator of Years of Life Lost (YOLL) is used to quantify the expected adverse effects on human health resulting from an increase in the concentration of air pollutants. The dose-effect models that are used here for impact assessment were recommended in ExternE (European Commission 1999) after a thorough review of the recent epidemiological literature. Following the ExternE recommendations, we assume a linear relationship between a change in mortality rate and the concentration of SO_2, ozone, primary fine particles (particles directly emitted from the stack), and secondary fine particles (sulphate and nitrate aerosols subsequently formed from the emission of gaseous SO_2 and NO_x). We do not consider direct negative effects on health from NO_x, because the epidemiological evidence is inconclusive. For a detailed discussion of the dose-effect models see European Commission (1999).

As this paper focuses on the analysis of spatial differentiation of effects, we do not consider non-fatal effects here, since we do not want to address the problem of aggregating different types of health effects in this context. Dose-effect models are available for a wide range of different health endpoints, and the aggregation of different morbidity effects by using monetary valuation is extensively discussed (European Commission 1999). The pattern of spatial distribution of non-fatal effects in general is similar to the distribution of mortality impacts. The omission of non-fatal effects here is by no means a value choice that suggests that one neglect the morbidity effects of LCIA.

In addition to the YOLL indicator, we use the accumulated exposure as an indicator for pressure on human health that

is definitively not biased by any assumptions or uncertainties related to effect modelling (see also Potting 2000). The accumulated exposure is defined here as the change in concentration resulting from a given emission, calculated for each grid cell of the modelling domain, multiplied with the number of persons living in the respective grid cells, and summed up over the modelling domain.

1.2.2 Acidification and terrestrial eutrophication

Our approach is based on the critical loads concept (e.g. Posch et al. 1995) developed in the frame of the UN-ECE Convention on Long-range Transboundary Air Pollution (United Nations – Economic Commission for Europe 1996). As a physical indicator, we take the change in ecosystem area protected against acidification and nutrifying nitrogen, respectively.

A critical load is defined as "a quantitative estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge" (United Nations – Economic Commission for Europe 1996). There exist several methods to derive critical loads. These methods differ with respect to the degree of sophistication and therefore to data requirements. Since we want to consider longer time horizons on a regional scale, the use of the so-called steady-state mass balance seems to be the most appropriate approach.

While critical loads for nutrifying nitrogen depend only on nitrogen deposition, both sulphur and nitrogen contribute to acidification, so that critical acid loads are given as so-called protection isolines. A protection isoline is a polygon for a given protected ecosystem area share built up by several pairs of nitrogen and sulphur deposition rates (for their derivation see e.g. Posch et al. 1995). This means if the actual deposition of nitrogen and sulphur is below the protection isoline, the corresponding area-fraction of ecosystems is, in the long term, considered to be protected against acidification.

The critical load data applied within our study are taken from (Posch et al. 1997). They are given for different levels of protected ecosystem areas (0, 5, 10,..., 85, 90, 91,..., 99, 100% of the grid cell's total ecosystem area) for each EMEP 150 x 150 km grid cell (the so called EMEP grid of the Cooperative Programme for Monitoring and Evaluation of the Long Range Transmission of Air Pollutants in Europe is extensively used in Europe for air quality modelling) covering the whole of Europe except for parts of Russia. The ecosystems for which critical acid load data are provided are semi-natural terrestrial and limnic, i.e. basically forests and lakes. The critical loads for nutrifying nitrogen are given for forest ecosystems. The change in protected ecosystem area per unit emission is calculated by comparing the resulting change in nitrogen and sulphur deposition grid cell-wise against the respective critical loads.

Country-specific acidification and terrestrial eutrophication factors have been presented before by other authors (Potting et al. 1998, Huijbregts 1999), i.e. the approach, as such, is not new. We present our results here to compare the impact factors derived from the EcoSense and the RAINS model.

1.2.3 Impacts on man-made environment

We have assessed direct effects of SO_2 and ozone on crop yield, and effects of SO_2 and wet acid deposition on building materials. For impact assessment, we again use the dose-effect models recommended by ExternE (European Commission 1999). In particular, the direct effects of SO_2 on crops strongly depend on background conditions. As sulphur is also a plant nutrient for agricultural crops, an increase in ambient SO_2 concentration leads to a positive effect on yield at low background concentrations, while there is a negative effect at higher background levels.

Table 1 summarises the crop species and the types of materials considered. Because of the large number of different crop species and types of materials, there is an obvious need for the aggregation of results. Although in this paper we want to avoid any discussion on the valuation of impacts, the use of monetary values for aggregation seems to be appropriate here, and less problematic than, for example, in the field of health effects, as we can use market prices for crops, and maintenance costs for materials.

Table 1: Effects on man-made environment included in the impact assessment

Impact category	Pollutant	Effects included
Material damage	SO_2, acid deposition	Ageing of galvanised steel, limestone, mortar, sandstone, paint, rendering, zinc for utilitarian buildings
Crops	SO_2	Yield change for wheat, barley, rye, oats, potato, sugar beet
	O_3	Yield loss for wheat, potato, rice, rye, oats, tobacco, barley, wheat

2 Modelling framework

For the modelling of the fate-exposure-effect chain, we use the integrated impact assessment model EcoSense. EcoSense was developed within the series of ExternE Projects on 'External Costs of Energy' funded by the European Commission (European Commission 1999). The model supports the quantification of environmental impacts by following a detailed site-specific 'impact pathway' approach, in which we try to model the causal relationships from the release of pollutants through their interactions with the environment to a physical measure of impact and, where possible, a monetary valuation of the resulting welfare losses. As it was the objective of the ExternE study to achieve an economic valuation of impacts, the impact assessment procedure is very much endpoint oriented, but it provides results on various intermediate levels of the environmental mechanism that can be used independently of any valuation methodology. The current version of EcoSense covers about 20 substances, including SO_2, NO_x, particles, NMVOCs (non-methane volatile organic compounds) and the relevant secondary pollutants, which are the most important pollutants for combustion processes, as well as some heavy metals and some organic substances. Although EcoSense was primarily designed for

assessing impacts from power plant emissions, the scope of the model was broadened to also cover other industrial and, in particular, transport activities.

A schematic flowchart of the EcoSense model is shown in Fig. 1. EcoSense provides harmonised air quality and impact assessment models together with a comprehensive set of relevant input data for the whole of Europe, which allow a site-specific impact assessment. A link to the European CORINAIR emission database allows the definition of emission scenarios by taking into account emission reduction measures in specific countries and industry sectors.

To cover different pollutants and different scales, EcoSense provides three air-quality models completely integrated into the system:

- *The Industrial Source Complex Model* (ISC) is a Gaussian plume model developed by the US-EPA (Brode and Wang 1992). The ISC is used for transport modelling of primary air pollutants on a local scale. Within EcoSense, the model is configured to calculate hourly concentration values of non-reactive substances for one year at the centre of 10 x 10 km grid cells, which are defined as a sub-grid of the 50 x 50 km EMEP grid. Effects of chemical transformation and deposition are neglected. Annual mean values are obtained by temporal averaging of the hourly model results.
- *The Windrose Trajectory Model* (WTM) (Trukenmüller et al. 1995) used in EcoSense to estimate the concentration and deposition of acid species on a regional scale was originally developed at Harwell Laboratory by Derwent and Nodop (1986) for atmospheric nitrogen species, and extended to include sulphur species by Derwent, Dollard and Metcalfe (1988). The model is a receptor-orientated Lagrangian plume model employing an air parcel with a constant mixing height of 800 m

moving with a representative wind speed. The results are obtained at each receptor point by considering the arrival of 24 trajectories weighted by the frequency of the wind in each 15° sector. The trajectory paths are assumed to be along straight lines and are started at 96 hours from the receptor point. Within EcoSense, WTM works on the EMEP 50 x 50 km grid.

- *The Source-Receptor Ozone Model* (SROM) integrated in the EcoSense package is based on source-receptor relationships (Simpson et al. 1997a). It is used to estimate ozone concentrations on a European scale. Input to SROM are national annual NO_x and anthropogenic NMVOC emission data from 37 European countries, while output is calculated for individual EMEP 150x150 km² grid squares by employing country-to-grid square matrices. To account for the non-linear nature of ozone creation, SROM utilises an interpolation procedure allowing source-receptor relationships to vary depending upon the emission level of the country concerned (Simpson et al. 1997b, Appendix B).

The current version of EcoSense includes a large number of exposure-response functions and monetary values that were compiled and thoroughly reviewed within the ExternE projects (European Commission 1995, 1999). The impact assessment modules calculate the physical impacts by applying the relevant exposure-response functions to each individual grid cell of a European wide grid (EMEP 50x50 km), taking into account the information on receptor distribution and concentration levels of air pollutants from the database. As EcoSense uses a full emission inventory for the modelling domain as a starting point for air quality and impact modelling, it is possible to analyse the contribution of a specific source of pollutants (or a source group) to the exceedance of thresholds (e.g. critical loads for ecosystems) by taking into account pre-defined background conditions.
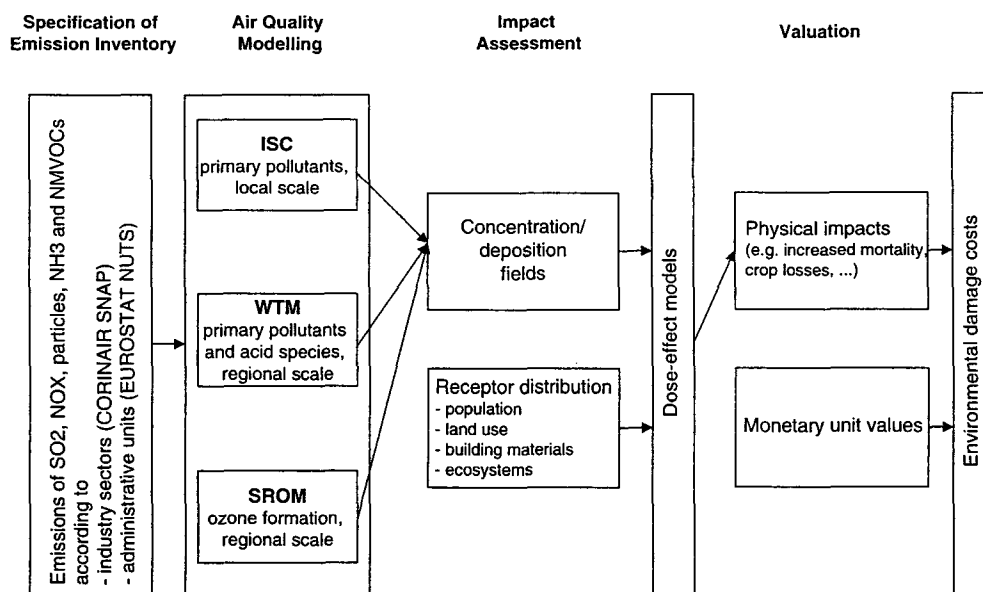

Fig. 1: Flowchart of the EcoSense model

While EcoSense was originally developed to assess environmental impacts from air pollution in Europe, we have recently finalised the implementation of (a reduced version of) the model in South America and Asia. These versions of EcoSense were used to calculate impact factors for some of the health effects, and to compare site dependent factors between Europe, South America, and Asia.

3 Results: country-specific damage factors

3.1 Health effects

Following the procedure described above, we have quantified effects on human health within the European population, expressed as accumulated exposure and as Years of Life Lost (YOLL), that result from a 10% increase of the respective national emissions of SO_2, NO_x, VOCs, and fine particles (PM_{10}), for all EU-15 countries (excluding Luxembourg). Model results were then normalised to a unit emission, so that the final damage factors can be presented as impact per tonne of pollutant emitted. **Fig. 2** exemplarily shows the spatial distribution of health impacts (YOLLs) in Europe resulting from a 10% increase of SO_2-emissions in Germany.

Country-specific damage factors are presented in **Table 2** (accumulated exposure per tonne of pollutant) and in **Table 3** (YOLL per kt of pollutant). As the underlying dose-effect models assume linear exposure-response functions without a threshold, the accumulated exposure can be transformed to Years of Life Lost by using a constant (substance specific) factor. For the further discussion of results, we will therefore consider only the YOLL indicator. Tables 2 and 3 show that, in general, there might be a large variation in impacts resulting from the same tonne of pollutant emitted both between source countries, and between the two different base years.
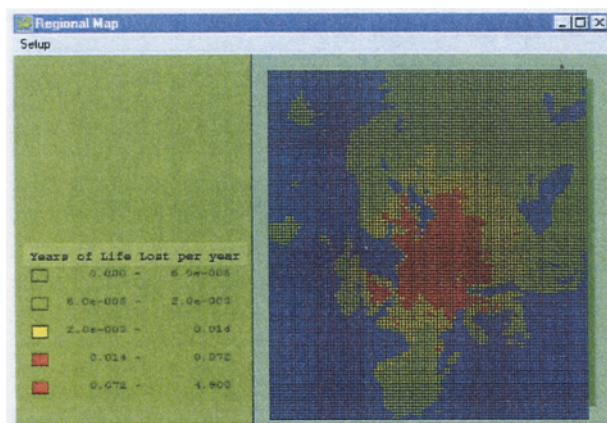

Fig. 2: Spatial distribution of health effects (YOLL) in Europe resulting from a 10% increase of SO_2-emissions in Germany

Within our model, we treat primary PM_{10} as non-reactive, and the exposure-response function linking PM_{10}-concentration to the change in mortality rate is assumed to be linear without a threshold. As a consequence, the impacts per unit PM_{10} emitted do not depend on the background concentration of PM_{10} in the atmosphere, so that the damage factors derived for PM_{10} are the same for 1990 and 2010. The differences between countries result from differences in the population distribution relative to the emission source, and from differences in meteorological conditions. The PM_{10}-damage factors differ by more than an order of magnitude between the EU-15 countries, ranging from 6 YOLL per kt PM_{10} emitted in Finland to about 90 YOLL per kt PM_{10} emitted in Belgium. Bearing in mind the population distribution in Europe, and the prevailing wind coming from the West, results are quite plausible: the effects per tonne of PM_{10} are lowest in the Scandinavian and Mediterranean coun-

Table 2: Accumulated exposure in the European population resulting from the emission of one tonne of pollutant from the respective source country (in Pers. µg/m³) (Base years 1990 and 2010)

	accum. O_3-exposure per tonne of NMVOC		accum. O_3-exposure per tonne of NO_x		accum. SO_2-exposure per tonne of SO_2		accum. Sulfate exposure per tonne of SO_2		accum. Nitrate exposure per tonne of NO_x		accum. PM_{10}-exposure per tonne of PM_{10}	
	1990	2010	1990	2010	1990	2010	1990	2010	1990	2010	1990	2010
Austria	126	165	-16	208	389	390	163	181	295	448	361	361
Belgium	288	348	-388	-289	601	601	176	187	215	416	590	590
Denmark	191	277	-77	125	161	162	73	77	131	167	143	143
Finland	53	91	68	131	58	58	27	29	49	54	39	39
France	186	204	23	138	439	442	175	186	334	455	401	401
Germany	265	280	-262	-40	407	403	132	158	189	393	437	437
Greece	56	56	31	31	158	163	99	94	219	188	207	207
Italy	173	162	-80	-18	283	252	119	115	227	320	306	306
Ireland	157	172	18	94	123	122	54	55	116	132	109	109
Netherlands	239	312	-429	-330	532	537	155	166	181	340	388	388
Portugal	120	119	9	10	172	172	73	74	140	149	155	155
Spain	79	85	32	80	161	163	95	101	180	214	210	210
Sweden	70	98	45	181	79	80	41	43	75	86	46	46
UK	197	238	-326	-255	333	333	92	101	117	189	258	258
EU-15	191	198	-157	-32	324	288	116	118	188	289	361	361

Table 3: Years of Life Lost (YOLL) within the European population resulting from the emission of one kilo-tonne of pollutant from the respective source country (Base years 1990 and 2010)

	YOLL per kt of NMVOC due to O_3 formation		YOLL per kt of NO_x due to O_3 formation		YOLL per kt of SO_2 due to exposure to SO_2		YOLL per kt of SO_2 due to formation of sulfate aerosols		YOLL per kt of NO_x due to formation of nitrate aerosols		YOLL per kt of PM_{10} due to exposure to PM_{10}	
	1990	2010	1990	2010	1990	2010	1990	2010	1990	2010	1990	2010
Austria	0.8	1.0	-0.1	1.3	2.1	2.1	36.8	44.4	44.0	69.6	56.5	56.5
Belgium	1.8	2.1	-2.4	-1.8	3.2	3.2	39.2	46.3	32.4	65.0	92.7	92.7
Denmark	1.2	1.7	-0.5	0.8	0.9	0.9	17.0	19.0	20.0	26.1	22.4	22.4
Finland	0.3	0.6	0.4	0.8	0.3	0.3	7.0	6.9	7.8	8.3	6.0	6.0
France	1.1	1.2	0.1	0.8	2.3	2.4	40.0	46.5	51.4	71.2	62.9	62.9
Germany	1.6	1.7	-1.6	-0.2	2.2	2.2	31.6	38.0	27.9	61.2	68.6	68.6
Greece	0.3	0.3	0.2	0.2	0.8	0.9	24.3	22.6	33.8	29.0	32.6	32.6
Italy	1.1	1.0	-0.5	-0.1	1.5	1.3	27.3	28.1	34.6	49.9	48.0	48.0
Ireland	1.0	1.1	0.1	0.6	0.7	0.7	12.7	13.6	17.8	20.7	17.1	17.1
Netherlands	1.5	1.9	-2.6	-2.0	2.8	2.9	34.9	40.8	27.4	53.2	61.0	61.0
Portugal	0.7	0.7	0.1	0.1	0.9	0.9	17.4	18.2	21.7	23.3	24.4	24.4
Spain	0.5	0.5	0.2	0.5	0.9	0.9	21.7	25.0	27.8	33.5	33.0	33.0
Sweden	0.4	0.6	0.3	1.1	0.4	0.4	9.6	10.4	11.5	13.4	7.3	7.3
UK	1.2	1.5	-2.0	-1.6	1.8	1.8	21.1	23.8	17.5	29.3	40.4	40.4
EU-15	1.2	1.2	-1.0	-0.2	1.7	1.5	27.0	28.7	28.5	45.1	56.7	56.7

tries, where the population density is relatively low, and a part of the particles is transported across the sea, while the central Western-European countries (Belgium, the Netherlands, Germany, France), from which particles are transported across heavily populated areas, show particularly high damage factors.

The situation is much more complex in the case of secondary pollutants, in particular for ozone. Ozone formation is a highly non-linear process which heavily depends on meteorological conditions as well as on the background concentrations of NO_x and VOCs, which in turn differ significantly over time and across different sites in Europe. The damage factors for NO_x acting via ozone formation given in Table 3 are shown again in **Fig. 3**. It can be concluded from Fig. 3 that the potential for ozone formation per tonne of NO_x increases from 1990 to 2010 due to the overall reduction of NO_x-emissions in Europe, even if VOC reductions are taken into account. However, the situation in the individual countries differ significantly from each other. There are several countries (Austria, Belgium, Denmark, Germany, Italy, the Netherlands, United Kingdom) in which the emission of an additional tonne of NO_x leads to a decrease in regional ozone concentration in 1990, which means that ozone formation is VOC limited. The emission reductions that are expected to take place until the year 2010 in some countries (Austria, Denmark) lead to a change of the sign of the damage factor. In Germany, the relatively large 'benefit' per tonne of NO_x is reduced to a small, but still positive effect (i.e. negative damage), while a tonne of NO_x emitted in Belgium, the Netherlands, or the United Kingdom still leads to a significant reduction of regional ozone in Europe. On the EU-15 average, an additional tonne of NO_x emitted under the conditions of the year 2010 still is expected to slightly reduce regional ozone in Europe.
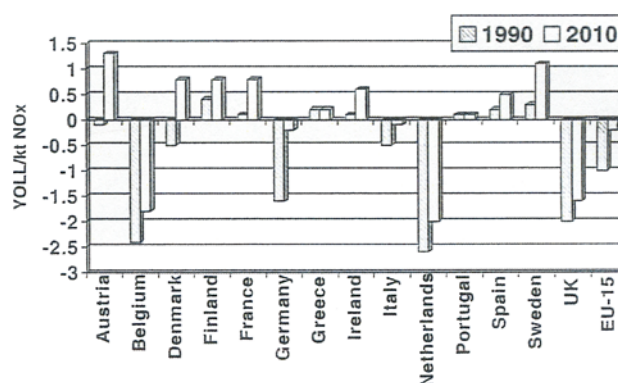
Fig. 3: Years of Life Lost per 1000 t of NO_x, acting via ozone formation

From these results we should of course not conclude that an additional tonne of NO_x emitted somewhere in Europe is necessarily something good for the environment. First of all, the effects from ozone seem to be small compared to other, more severe impacts resulting from NO_x-emissions (see below). Secondly, it is expected that a further reduction of ozone precursor emissions that are anticipated to take place after 2010 will result in a situation in which an additional unit of NO_x emission from any of the EU-15 countries will lead to an increase of ozone concentration and thus to a real damage. However, results are consistent with our understanding of ozone formation and should not be considered as a modelling artefact. Taking into account the level of NO_x and VOC emissions in Europe in the years 1990 and 2010 as a working point, a marginal increase in NO_x emissions in some countries (and on the EU-15 average) leads to a reduction of adverse health effects from ozone.

Under the conditions of the two base years analysed here, the influence of background conditions on the formation of ozone resulting from NMVOC emissions is less significant than for NO_x (Table 3). The NMVOC damage factors for the EU-15 average are the same for the years 1990 and 2010 (1.2 YOLL per kt of NMVOC), but the variation between countries still reaches a factor of 7 (0.3 YOLL per kt NMVOC from Greece, 2.1 YOLL per kt NMVOC from Belgium).

Fig. 4 shows the damage factors for NO_x acting through the formation of secondary fine particles (nitrate aerosols). The formation of nitrate aerosols depends on the ratio of the $SO_2:NO_x:NH_3$ concentration in the atmosphere, which is different between countries, but also changes from 1990 to 2010 because of the non-proportional reduction of the three pollutants. It is obvious that in this case the effects per tonne of NO_x are much more severe than the partly positive impacts on health via ozone formation, so that in total we have a real damage per tonne of NO_x-emission for all countries. The variation of the damage factors between countries is close to an order of magnitude. Like in the case of ozone formation, the potential for the formation of nitrate aerosols and thus for adverse effects on human health from the same tonne of NO_x is expected to be increasing from 1990 to 2010.
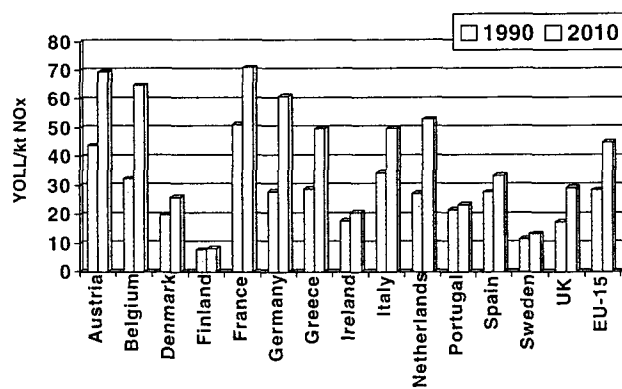

Fig. 4: Years of Life Lost (YOLL) per 1000 t of NO_x acting via the formation of nitrate aerosols

The degradation of SO_2 in the atmosphere due to chemical transformation is only slightly influenced by background conditions, so that the damage factors for SO_2 – acting directly and via the formation of sulphate aerosols – do not differ very much between the two base years (Table 3). The variation between countries is again one order of magnitude for the direct effects of SO_2, and about a factor of 7 for the effects via sulphate aerosol formation.

From **Table 4**, we learn that the damage factors derived for specific source sectors on the EU-15 average do not differ very much from the EU-15 average factors given in Table 3. This low variation between the different source sectors can be explained by the fact that activities within the rather unspecific main source categories are homogeneously distributed across the EU-15 countries. More significant differences between sector specific damage factors are expected when looking at more specific activities, for example, like the paper-mill industry as a sub-sector of industrial production processes, as the spatial pattern of industrial activities is more distinct on a lower aggregation level. Damage factors for more specific industrial activities can be derived easily from additional model runs. Because of the small differences between the human health damage factors for the main source sectors, we do not present sector specific factors for the other impact categories in the following sections.

The comparison of average damage factors derived for Europe, Asia, and South America shows that the differences between continents are smaller than the differences between individual countries of a continent (Table 5). On the continental average, one of the main parameters determining the health-related damage factors is the population density, which, for the three continents considered, is lowest in South America, and highest in Asia. However, the population density does not fully explain the differences between the continental damage factors, parameters like average wind speed, wind direction, and the location of main urban centres also have a significant influence.

Table 4: Years of Life Lost (YOLL) within the European population resulting from the emission of one kilo-tonne of pollutant from different source sectors in Europe (Base year 2010)

|	YOLL per kt of NMVOC due to O_3 formation	YOLL per kt of NO_x due to O_3 formation	YOLL per kt of SO_2 due to exposure to SO_2	YOLL per kt of SO_2 due to formation of sulfate aerosols	YOLL per kt of NO_x due to formation of nitrate aerosols	YOLL per kt of PM_{10} due to exposure to PM_{10}
Public power, cogeneration, and district heating plants	1.4	- 0.4	1.5	27.9	40.0	61.8
Commercial, institutional, and residential combustion plants	1.2	- 0.2	1.7	31.6	52.7	52.4
Industrial combustion	1.2	- 0.2	1.6	30.2	47.6	54.5
Production processes	1.7	0.1	1.7	30.6	40.6	54.7
Extraction and distribution of fossil fuels	1.3	- 1.5	1.9	41.3	27.3	
Solvent use	1.3	n. r.	n. r.	n. r.	n. r.	n. r.
Road transport	1.3	- 0.2	1.8	34.6	47.9	52.6
Other mobile sources and machinery	1.2	0.3	1.2	25.2	67.7	44.3
Waste treatment and disposal	1.0	- 0.3	1.8	36.2	46.9	59.5
Agriculture	1.0	- 0.4	n. r.	n. r.	51.4	47.4
n. r. : not relevant for source sector						

Table 5: Comparison between continents – Years of Life Lost (YOLL) resulting from the emission of one kilo-tonne of pollutant (Base year 1990)

	YOLL per kt of SO_2 due to exposure to SO_2	YOLL per kt of SO_2 due to formation of sulfate aerosols	YOLL per kt of NO_x due to formation of nitrate aerosols	YOLL per kt of PM_{10} due to exposure to PM_{10}
EU-15 average	1.7	27.0	45.1	56.7
Asia average	2.5	55.2	56.9	130.8
China	4.6	104.7	145.2	131.7
Japan	2.5	36.1	39.7	84.5
South Korea	3.5	50.3	47.6	101.0
South America average	0.34	4.9	6.8	16.3
Brazil	1.2	13.3	10.9	16.4
State of Sao Paulo	3.9	38.5	52.5	39.9
Colombia	0.33	3.6	6.0	5.5

3.2 Acidification and terrestrial eutrophication

Considering the critical loads for acidity, we can estimate additional unprotected ecosystem areas per tonne of pollutant emitted as given in Table 6. For the different emission scenarios of 1990 and 2010, respectively, the damage factors vary to an order of magnitude between the countries, especially for SO_2. Under the 1990 emission scenario, the Scandinavian countries show the highest, and the Mediterranean countries (Portugal, Greece, Spain) show the lowest damage factors. A comparison between both base years shows that there is no uniform shift in the damage factors. Emission reductions expected until 2010 lead, for example, to a remarkable decrease of the Finnish damage factors, and to a significant increase in the damage factors for Germany (Fig. 5). Fig. 6 shows schematically that these different trends over time can be explained by the step-function like shape of the different protection isolines, and by different background conditions between countries and base years.

Regionalised impact factors for acidification have already been calculated by Potting et al. (1998) and Huijbregts (1999). A comparison of our aggregated results for limnic and terres-
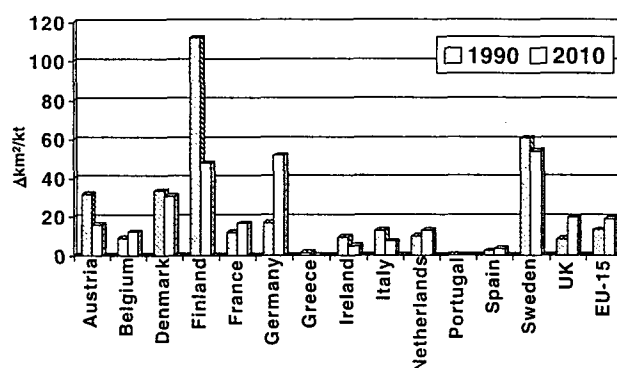
Fig. 5: Acidification – change in critical loads exceedance area per 1000 t of SO_2

trial ecosystems with the results from Potting et al. shows remarkable differences. In 1990, our results yield between 2 to 14 times higher damage factors for Greece, Italy, Austria, and Spain and lower damage factors especially for the United Kingdom and Sweden (less than 50% of Potting's values). Under the 2010 emission scenario, our damage factors for Portugal,

Table 6: Acidification – change in unprotected ecosystem area per unit emission (km^2/kt)

|	Aquatic ecosystems				Terr. ecosystems			
	SO_2		NO_x		SO_2		NO_x	
	1990	2010	1990	2010	1990	2010	1990	2010
Austria	5.9	0	2.4	0	25.8	15.9	7.5	6.2
Belgium	1.8	1.1	1.6	0.7	7.0	11.0	3.1	6.0
Denmark	18.1	14.5	9.8	2.6	15.2	16.3	7.1	3.4
Finland	35.5	19.4	9.9	0	76.3	28.3	24.0	6.3
France	2.1	0.8	0.9	0.4	9.9	15.8	3.5	9.5
Germany	4.5	7.9	1.9	2.9	12.5	43.7	6.5	18.2
Greece	0.7	0	0.8	0	0.7	0	0.9	0
Ireland	2.4	0	3.6	0	7.0	5.0	7.9	2.2
Italy	0.3	0	0.2	0.1	12.5	7.3	6.0	9.3
Netherlands	2.8	1.9	4.6	1.2	7.2	11.0	5.4	5.7
Portugal	0	0	0	0	0.2	0.4	0	0.4
Spain	0.2	0	0.3	0	2.1	3.5	1.1	3.0
Sweden	29.8	24.3	8.7	4.0	30.7	29.4	7.4	7.0
UK	2.3	4.1	0.9	1.6	6.2	15.5	2.8	8.3
EU-15 average	3.2	2.9	1.8	1.3	9.9	15.9	5.5	10.6
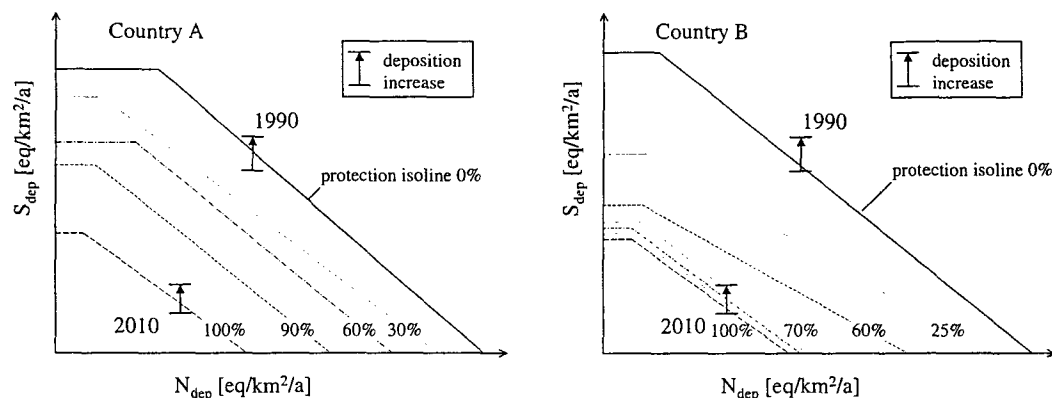
Fig. 6: Change in the protection levels in two countries caused by a unit deposition increase under different background conditions (schematic example: Country A: unit deposition increase in 1990 → 30% increase of unprotected area, in 2010 → 10%; Country B: unit deposition increase in 1990 → 25% increase of unprotected area, in 2010 → 40%)

Spain, and Italy as well as Germany are between 2 and 5 times higher than those of Potting et al. Lower damage factors are found especially for Greece and Ireland.

Rather than calculating risks of actually exceeding critical loads (Potting et al. 1998, and this study), Huijbregts follows a different kind of site-dependent characterisation of environmental impact. He considers accumulated relative risks that are normalised to the Swiss emissions. Since Huijbregts follows a different approach, a comparison between his and our results for acidification is not directly feasible. For comparing his results with those of Potting et al., Huijbregts recalculated both impact scores to acidification potentials referenced to SO_2. Compared to acidification potentials that are not based on fate and effect modelling (Heijungs et al. 1992), our acidification potentials of NO_x are of the same order as those of Potting et al. (1998) and Huijbregts (1999) (Table 7).

Table 7: Comparison of EU-average acidification potentials

	Our findings		(Potting et al. 1998)[a]		(Huijbregts 1999)[b]		(Heijungs et al. 1992)
	1990	2010	1990	2010	1990	2010	
NO_x	0.28	0.32	0.36	0.23	0.34	0.29	0.7
SO_2	1	1	1	1	1	1	1

[a] taken from (Huijbregts 1999); [b] only above-thresholds scenarios (OA)

The variation in results between the three studies are due to several differences with regard to the input data and modelling approach:

- Emission data: For most of the countries, the difference between our emission data for 1990 and the respective emission data used by Potting et al. and Huijbregts is less than ±10%, and is less than ±50% for all countries. The variation, however, is larger for the year 2010, reaching up to ± 60%, with an extreme of -135% difference between the SO_2-emissions in Ireland in our data and the data used by Potting et al. These differences lead to a different absolute change in emissions in the respective model runs (10% change of the country's total emissions), and to different background concentration levels, which both have a strong influence on the damage factors.
- Critical load data: All three studies use critical load data that were published by the UN ECE Centre for Effects at

RIVM in different years (Potting et al. from 1995, our study from 1997, Huijbregts from 1999). We have not yet analysed the influence of the different data sets on the damage factors. In addition, the way critical load data are used to derive unprotected ecosystem areas differs between the studies. Potting et al. and our study use protection isolines, whereas Huijbregts uses maximum critical acid load values for N and S that lead to an underestimation of the potential impact of NO_x or SO_2 emissions.
- Air quality modelling: The models used differ with respect to the meteorological data (different base years) and the transport modelling concept (country to grid matrices based on a trajectory model with a resolution of 150 x 150 km in RAINS vs. grid to grid trajectory model with a resolution of 50 x 50 km in EcoSense).

Because of the large number of different parameters, a detailed comparison of the different approaches is beyond the scope of the present paper. A more detailed inter-comparison of the models and basic assumptions, however, is desirable to increase the reliability and usefulness of the results.

Considering the critical loads for nutrifying nitrogen, we estimated additional unprotected ecosystem areas per tonne of pollutant emitted as given in Table 8. The damage factors

Table 8: Eutrophication of terrestrial ecosystems – change in unprotected ecosystem area per kt of NO_x (km²/kt)

	1990	2010
Austria	18.7	35.9
Belgium	8.7	16.7
Denmark	27.4	15.9
Finland	94.4	211.9
France	14.5	29.5
Germany	15.4	32.1
Greece	16.9	23.6
Ireland	12.2	14.4
Italy	18.5	29.5
Netherlands	12.3	19.1
Portugal	28.6	42.2
Spain	22.5	35.7
Sweden	47.6	40.6
UK	8.9	12.4
EU-15 average	18.8	37.7

Table 9: Impacts on crop yield and building materials. Damage costs resulting from the emission of one kilo-tonne of pollutant from the respective source country (Base years 1990 and 2010)

	Impact on crop yield						Impact on building materials			
	Euro per t of NMVOC due to O_3 formation		Euro per t of NO_x due to O_3 formation		Euro per t of SO_2 due to exposure to SO_2		Euro per t of SO_2		Euro per t of NO_x	
	1990	2010	1990	2010	1990	2010	1990	2010	1990	2010
Austria	455	497	312	871	27	-45	164	187	101	157
Belgium	829	938	-660	-263	-9	-87	117	134	45	87
Denmark	585	687	-22	397	-1	-34	74	82	42	55
Finland	99	69	144	129	-5	-8	25	26	13	13
France	581	537	528	725	-11	-63	91	106	59	82
Germany	834	788	-348	370	58	-62	123	142	53	108
Greece	512	470	900	814	-1	-7	70	68	58	46
Italy	1479	1063	-213	403	3	-14	74	74	58	81
Ireland	219	222	48	108	1	-8	27	33	21	24
Netherlands	642	834	-716	-329	-6	-77	110	127	42	77
Portugal	564	504	638	575	-4	-7	27	30	18	19
Spain	293	263	417	490	-4	-13	30	37	23	28
Sweden	164	161	184	257	-1	-12	37	40	25	29
UK	390	464	-406	-166	13	-30	60	72	23	35
EU-15	723	617	-79	321	20	-31	81	79	43	65

vary to an order of magnitude between the countries for the different emission scenarios of 1990 and 2010, respectively. Comparing both years, there is a uniform shift in the damage factors to higher values in 2010 except for Denmark and Sweden. In both years, the highest damage factors are calculated for Finland and the lowest for the United Kingdom, Belgium, the Netherlands, and Ireland.

3.3 Man-made environment (crop yield and building materials)

The effects of a unit change in emissions of SO_2, NO_x, and NMVOCs on crop yield and building materials are summarised in **Table 9**. The impacts on different crop species and different types of building materials are aggregated into a single indicator (damage costs) by using crop market prices and maintenance costs for materials (European Commission 1999). Results show that there is a particularly large variation in the effects of SO_2 and NO_x (via ozone formation) on crop yield both between countries and base years. Like in the case of health effects, there is a negative damage (i.e. a benefit) per tonne of NO_x for several countries due to the effect of NO_x emissions on the regional ozone concentration. The change in background conditions between 1990 and 2010 leads to a change in the sign of the damage factor for several countries (including the EU-15 average).

As discussed above, depending on the background concentration, SO_2 might have either a positive or a negative effect on crop yield. While in 1990 an additional tonne of SO_2 still caused a reduction in crop yield on the EU-15 average, the expected SO_2 emission reduction lead to a significantly reduced SO_2 background concentration in 2010, a condition under which additional sulphur is expected to cause an increase in crop yield.

The effects of NMVOCs on crops (via ozone formation) and the effects of SO_2 and NO_x on materials per unit emission do not differ very much between the two base years, but there is a significant variation between the EU-15 countries.

4 Discussion and Conclusions

In the previous sections, we presented country-specific damage factors for different impact categories and pollutants, which show different levels of variation between countries, and between base years. To better understand the reasons for these differences and their potential implications for LCA applications, it is helpful to group different mechanisms of action into different categories according to their respective sensitivity towards site-dependent parameters. Taking into account the pollutants and effects considered in the previous analysis, we can basically differentiate between four major categories:

I) Non-reactive pollutants, linear dose-effect model without threshold
Example: mortality from primary (i.e. directly emitted) fine particles
The impact per unit emission does not depend on the level of background concentration. Differences between site dependent damage factors are determined by the distribution of receptors (e.g. population) relative to the emission source, and the relevant meteorological conditions. Results show that the variation between country-specific damage factors for EU-15 countries reach an order of magnitude. The difference between the average damage factors for different continents (Europe, Asia, South America) is smaller.

II) **Non-reactive pollutant, non-linear dose-effect model**
Example: Effects of SO_2 on crops
Impacts per unit emission differ between sites because of receptor distribution and meteorological conditions, and because of changing levels of background concentrations. Results show that there might be a change in the sign of the damage factor between different base years.

III) **Secondary pollutant (non-linear formation), linear dose-effect model without threshold**
Example: Effects of ozone on human health
In contrast to case (II), the effect model here is assumed to be linear, while the formation of the acting pollutant is highly non-linear. Like in (II), we observe a change in the sign of the damage factors both between countries, and between base years.

IV) **Secondary pollutants, non-linear dose-effect model (threshold)**
Example: Exceedance of critical loads for acidification
Both the formation of the acting agent, and the expected impact depend on the level of background concentration. Change in the damage factors due to changes in regional background conditions might go into different directions for different countries.

For the quantification of acidification and eutrophication impacts, we followed the approach already suggested by Potting et al. (1998), which has been criticised because it does not properly reflect the 'consumption' of a system's buffer capacity by emissions that do not contribute to the exceedance of a threshold. We share this concern, and in the tradition of LCA thinking it seems to be desirable to put a weight on any unit of emission that might contribute to a potential (future) impact. However, as soon as we make an attempt towards endpoint modelling and weighting across different endpoints, the actual marginal impact seems to be the most relevant measure characterising the effect that raises societal concern.

The same consideration applies not only to the 'simple' threshold problem, but to all the non-linear mechanisms belonging to categories (II) and (III). We could use our models to derive non-marginal damage factors by comparing impacts from e.g. an emission scenario for the year 2010 against impacts resulting from a pre-industrial concentration level of pollutants, but it is questionable to what extent such results provide good guidance to decisions which have to take into account today's environmental conditions.

A major problem for the usefulness and reliability of the acidification/eutrophication results is the fact that the differences between the results from different studies are in the same range as the differences between countries in the respective studies, indicating that results should be used with great care only. There is an obvious need for carrying out a thorough cross-comparison of input data, assumptions and modelling approaches to provide a more consistent picture in the future.

The variation in the damage factors derived for the main CORINAIR source sectors is lower than we originally expected. A more detailed analysis of more specific industrial activities will most likely lead to a higher variation between sectors. However, the driving forces for this variation are site dependent characteristics rather than the affiliation to a specific source sector, which for various reasons might be clustered around specific locations.

How to deal with negative impact factors? They certainly provide helpful information for environmental priority setting, suggesting e.g. that it is probably more effective to reduce VOC emissions rather than NO_x emissions currently to combat ground level ozone in Europe. But – again – we should not interpret negative factors in a way saying that emitting more is beneficial for the environment, and thus we do not want to suggest the introduction of negative characterisation factors to be used in LCA. In the case of negative damage factors, it might be defendable to set characterisation factors to zero, thus indicating that the potential for environmental damage is low in a specific context.

The application of country-specific characterisation factors (presented in a set of look-up tables) in life cycle impact assessment seems to be straightforward. Depending on the scope of the study, and the information available from the life cycle inventory, regional (continental) and country-specific factors can be used as a complement to global characterisation factors. The partly large variation between country-specific impact factors indicates that the consideration of site dependent effects can help to make the results of a life cycle impact assessment more meaningful.

From our point of view, the influence of background conditions changing over time is more difficult to capture in LCIA than the influence of site dependent parameters. Our results show that changes in background conditions within a relatively short time period can lead to significant changes in the damage factors, which causes problems when dealing with processes that emit over a 'long' lifetime (more than five or ten years). The prediction of future background conditions is more uncertain than the analysis of site-dependent parameters under given conditions, and the allocation of emissions to specific years might be problematic. Additional model runs can of course be used to produce a time series of damage factors, but we consider the results derived for the year 2010 as a reasonable proxy for the conditions of the next years.

We consider the results of the present paper as a further step towards the integration of site-dependent modelling into life cycle impact assessment, and also as a step towards the integration of different modelling disciplines. We certainly do not want to question the usefulness and validity of site-independent LCIA, but, on the other hand, our results show that temporal and spatial aspects may be important. However, there is still a very long way to go to establish a comprehensive set of site dependent characterisation factors. In our study, we carried out model runs for a limited set of air pollutants, and for a limited set of countries. The different versions of the EcoSense model (Europe, Asia, South America) cover more countries than those exemplarily analysed in the present paper, so that additional model runs can provide a more comprehensive set of country-specific data. A priority for the near future is to increase the number of sub-

stances covered. We have started work on the integration of heavy metals and persistent organic pollutants into the European version of EcoSense.

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